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# Ecological Effects of Climate Change on Salt Marsh Wildlife: A Case Study from a Highly Urbanized Estuary

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**ABSTRACT**

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Coastal areas are high-risk zones subject to the impacts of global climate change, with significant increases in the frequencies of extreme weather and storm events, and sea-level rise forecast by 2100. These physical processes are expected to alter estuaries, resulting in loss of intertidal wetlands and their component wildlife species. In particular, impacts to salt marshes and their wildlife will vary both temporally and spatially and may be irreversible and severe. Synergistic effects caused by combining stressors with anthropogenic land-use patterns could create areas of significant biodiversity loss and extinction, especially in urbanized estuaries that are already heavily degraded. In this paper, we discuss current ideas, challenges, and concerns regarding the maintenance of salt marshes and their resident wildlife in light of future climate conditions. We suggest that many salt marsh habitats are already impaired and are located where upslope transgression is restricted, resulting in reduction and loss of these habitats in the future. In addition, we conclude that increased inundation frequency and water depth will have negative impacts on the demography of small or isolated wildlife meta-populations as well as their community interactions. We illustrate our points with a case study on the Pacific Coast of North America at San Pablo Bay National Wildlife Refuge in California, an area that supports endangered wildlife species reliant on salt marshes for all aspects of their life histories.

**ADDITIONAL INDEX WORDS:** *California, California clapper rail, coastal, conservation, endangered, salt marsh harvest mouse, San Francisco Bay, sea-level rise, storms.*

## INTRODUCTION

Global warming scenarios include projected changes in mean and extreme ambient temperatures, precipitation and seasonal patterns, ocean temperature and acidity, and extreme climatic events and sea-level rise (Cayan *et al.,* 2005; Hansen *et al.,* 2006; Harris *et al.,* 2006; IPCC, 2007). Coastal ecosystems are particularly sensitive to the impacts of climate change because they represent a narrow, transitional ecotone between the marine and terrestrial environments. They have been designated as high-risk zones subject to climate change impacts and are especially sensitive to: (1) projected sea-level rise, (2) ocean temperatures, and (3) global atmospheric and oceanic circulation patterns resulting in an increase in storm frequency and intensity (Cayan *et al.,* 2008a; Hoegh-Guldberg and Bruno, 2010; IPCC, 2007).

Global sea-level rose 1.8 mm/y between 1961 and 1993, and it has risen 3.1 mm/y since 1993 (IPCC, 2007). Projections of mean sea-level rise by 2100 are characterized by high uncertainty due to the difficulty in modeling melting ice-sheet dynamics and other processes not included in earlier modeling efforts (IPCC, 2007). While earlier sea-level rise projections ranged from 0.19 to 0.58 m (IPCC, 2007), more recent projections are closer to 0.6–1.6 m (Jevrejeva, Moore, and Grinsted, 2010) or 0.9–1.3 m (Grinsted, Moore, and Jevrejeva, 2010) by 2100. Vermeer and Rahmstrof (2009) projected a rise in sea level of 1.9 m by 2100 contingent upon ambient temperature conditions. Warmer sea-surface temperatures have increased the number and proportion of category 4–5 hurricanes since 1970 (Emanuel, 2005; Webster *et al.,* 2005), posing significant threats to coastal areas from storm surges, sustained winds, and large amounts of rainfall in a short time period (Mousavi *et al.,* 2011). Bender *et al.* (2010) and Knutson *et al.* (2010) showed that by the end of the twenty-first century, an increase in the average intensity of extreme cyclone events will occur. Increased sea level (Nicholls and Cazenave, 2010) and increased storm frequency and/or intensity that coincide with high tides are the greatest threat to the near-term sustainability of coastal ecosystems (Cayan *et al.,* 2008a).

Tidal salt marshes are highly productive ecosystems found in the terrestrial-marine ecotone (Archibold, 1995; Mitsch and Gosselink, 2000). Sea-level rise and an increase in extreme climate events will be the most significant factors threatening

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these ecosystems and their dependent biodiversity (Cayan *et al.,* 2008a; Craft *et al.,* 2009; FitzGerald *et al.,* 2008; IPCC, 2007; Kirwan *et al.,* 2010; Menon *et al.,* 2010). In addition, an increase in ambient temperature has been shown to cause vegetation dieback in salt marshes (McKee, Mendelssohn, and Materne, 2004); however, impacts from ocean acidification and ocean temperature changes on estuaries are highly uncertain (Hoegh-Guldberg and Bruno, 2010).

## TIDAL SALT MARSH ECOSYSTEMS

Salt marshes are found along low-energy coastlines at mid- to high latitudes (Mitsch and Gosselink, 2000). Occurring in both the Northern and Southern Hemispheres, the geographical extent of tidal salt marshes is primarily within the intertidal zones of temperate estuaries (16,000 km$^2$ in North America). They are also present in the subarctic boreal ecosystems but are replaced by mangrove ecosystems in tropical regions (Greenberg *et al.,* 2006; Jefferies, 1977). Salt marshes are dominated by halophytic plants with tolerance for high salinity levels and tidal inundation, resulting in strong zonation of vegetation from lower to higher elevations (Mancera *et al.,* 2005). The distribution of vegetation types can be closely related to channel network configuration and tidal drainage patterns (Sanderson, Ustin, and Foin, 2000), and their geomorphology may be shaped over hundreds to thousands of years (Goman, Malamud-Roam, and Ingram, 2008; Willard, Cronin, and Verardo, 2003).

Human settlement and development are ongoing threats to salt marshes because they are geographically limited. Globally, more than 600 million people, or 10% of the world's population, live along coastlines between 0 and 10 m in elevation (McGranahan, Balk, and Anderson, 2007). In the United States alone, an estimated 45% of the population resides in coastal areas (Culliton *et al.,* 1990). Anthropogenic land alterations such as filling, diking, and draining to convert salt marshes to solar salterns, agricultural lands, and urban development have significantly impacted their distribution around the world (Atwater *et al.,* 1979; Kennish, 2001).

Since the melting of the late Pleistocene ice sheets and the Flandarian transgression of sea level, ca. 6000 YBP, salt marshes have kept pace with sea-level changes in a period of decelerated sea-level rise (Stanley and Warne, 1994). The resiliency and persistence of a salt marsh with sea-level rise are context-specific and vary by location and where a site lies within the tidal range (Kirwan and Guntenspergen, 2010; Warren and Niering, 1993). With sea-level rise, salt marshes can still be self-sustaining provided they have sufficient sediment input and time to respond (Reed, 2002). Tidal action transports suspended sediment in salt marshes via channel networks and along open-water edges to create dynamic geomorphic systems (Reed *et al.,* 1999). In addition, organic inputs and belowground biomass have been shown to be important contributions to salt marsh development and accretion (Drexler, de Fontaine, and Brown, 2009; Goodman, Wood, and Gehrels, 2007), potentially helping the salt marsh keep up with sea-level rise.

Accretion rates for salt marshes situated on the Pacific Coast of North America have kept pace with sea-level rise in the last 500–2000 y (Goman, Malamud-Roam, and Ingram, 2008; Watson, 2004). However, watershed damming and underground extractions can inhibit processes that maintain salt marsh elevation in relation to sea level (Ravens *et al.,* 2009; White and Tremblay, 1995; Wilson and Allison, 2008). Salt marshes in southwestern Louisiana and Galveston Bay, Texas, are subsiding due to insufficient sediment supply and underground water and oil extraction (Ravens *et al.,* 2009; Rybczyk and Cahoon, 2002; Wilson and Allison, 2008).

On the Atlantic seaboard of North America, plant species found in the low marsh have already migrated to upper marsh zones, resulting in altered plant communities with recent changes in sea level and storms (Donnelly and Bertness, 2001; Warren and Niering, 1993). Projected sea-level rise and more frequent extreme storm events will alter sediment inputs and distribution patterns, changing hydraulic flow dynamics (Culberson, Foin, and Collins, 2004; Day *et al.,* 2008; FitzGerald *et al.,* 2008; Morris *et al.,* 2002). As a result, the distribution of plant communities will shift. In addition, an increase in flooding frequency of upland salt marsh areas and saltwater intrusion over levees into adjacent lands may occur during extreme events (FitzGerald *et al.,* 2008; French, 2006). The coupling of storm surges and high tides will result in erosion of the salt marsh edge and excessive flooding of vegetation (Erwin, Sanders, and Prosser, 2004; Evens and Page, 1986; Zedler, 2010).

Regardless of which climate change projection and resultant sea-level rise and storm model is used through 2100, thousands of hectares of coastal salt marshes will be permanently inundated if accretion processes are not able to keep up. If natural or anthropogenic barriers inhibit salt marsh transgression to higher elevations, even greater loss will occur, often referred to as "coastal squeeze." In this paper, we categorize the climate change effects on salt marshes and their biota (Table 1) into short- ($\leq$25 y) and long-term ($>$25 y) effects.

## WILDLIFE SENSITIVITY AND ADAPTABILITY

Studies have shown that wildlife populations in many ecosystems around the world are already responding to climate change effects (Parmesan, 1996, 2006; Parmesan *et al.,* 1999; Previtali *et al.,* 2009; Solonen, 2008; Thomas and Lennon, 1999). Sensitivity and adaptability of wildlife to climate change effects will depend on local rates of change and the spatial habitat patterns. Wildlife species facing new environmental conditions (*e.g.,* beyond the range of historical variability) respond through changes in phenology, shifts in geographic distribution, adaptation through evolution or phenotypic plasticity, demographic structure change, or local extinctions (Austin and Rehfisch, 2003; Hoegh-Guldberg, 1999; Parmesan and Yohe, 2003; Brierley and Kingsford, 2009; Crimmins *et al.,* 2011). Species such as the polar bear (*Ursus maritimus*) that are highly dependent on specific habitat types (Derocher, Lunn, and Stirling, 2004) will be more vulnerable. Many species will react independently and be affected at different rates according to their ecological and physiological constraints (Root and Schneider, 2006; Willis *et al.,* 2010).

Table 1. *Project change, short-term impacts to wildlife, long-term impacts, and associated references. Impacts to salt marsh wildlife will vary temporally and locally. Some impacts will be to the wildlife species themselves, whereas others will be on their vegetative habitats. Here, short-term impacts are defined as measurable changes in the next 25 y, whereas long-term impacts are beyond 25 y* (U.S. Department of Interior and Council on Environmental Quality, 2012).

| Projected Change | Short-Term Impacts (≤25 y) | Long-Term Impacts (>25 y) | Example References |
|---|---|---|---|
| Increase in mean sea level | Drowning<br>Nest or young failure<br>Increase in predation<br>Flooding of available habitat<br>Migration or displacement of animal<br>Population decline | Extirpation<br>Reorganization of communities<br>Adaptation of species<br>Range shifts of vegetation<br>Range shifts of animals | Austin and Rehfisch (2003)<br>Menon *et al*. (2010)<br>Rush *et al*. (2009)<br>Seavey, Gilmer, and McGarigal (2011) |
| Increase in ambient temperature | Vegetation heat stress<br>Increased primary productivity<br>Animal heat stress<br>Decrease in reproduction<br>Animal metabolic rate increase | Extirpation<br>Phenology changes in salt marsh habitat community<br>Range shifts of habitat vegetation<br>Range shifts of animals | Gilg, Sittler, and Hanski (2009)<br>Hansen (2009)<br>Kirwan, Guntenspergen, and Morris (2009)<br>Lawler *et al*. (2009)<br>Stralberg *et al.* (2009)<br>Thomas and Lennon (1999) |
| Increase in storm frequency and intensity | Drowning<br>Nest or young failure<br>Increase in predation<br>Impacts to vegetation structure<br>Habitat loss (temporary)<br>Population decline | Extirpation<br>Scouring of salt marsh areas<br>Range shifts<br>Plant deaths<br>Habitat constriction (permanent) | Cayan *et al.* (2008a)<br>Hopkinson *et al.* (2008)<br>Meehl *et al.* (2000)<br>Parmesan *et al.* (2000)<br>Previtali *et al*. (2009)<br>Zedler (2010) |
| Changes in seasonal freshwater input | Seasonal drowning<br>Nest or young loss<br>Vegetation changes | Phenology change<br>Vegetation community composition change<br>Vegetation range shifts | Knowles and Cayan (2002)<br>Smith *et al*. (2009) |
| Ocean acidification | Unknown | Potential loss of food web components | Hoegh-Guldberg (1999)<br>O'Donnell *et al*. (2010) |

Geographic range and phenological shifts, as well as behavioral changes, have great potential to alter the distributions of populations and produce new communities through reassembly of species associations (Hughes, 2000; Root and Schneider, 2006; Schneider and Root, 2002; Stralberg *et al.,* 2009). Phenological shifts of migratory songbirds have been documented in both hemispheres, indicating that species are capable of advancing their spring arrival dates (Beaumont, McAllan, and Hughes, 2006; Macmynowski *et al.,* 2007; Parmesan, 2007). Different phenological responses across trophic levels may lead to a mismatch between interrelated populations, such as the European great tit (*Parus major*) and their insect prey (Visser *et al.,* 1998). Changes in latitudinal and altitudinal geographical distribution have been documented for both mammals and birds (Moritz *et al.,* 2008; Root *et al.,* 2003), ranging from American pikas (*Ochotona princeps*; Beever *et al.,* 2011) to hummingbirds (Buermann *et al.,* 2011). However, such range shifts are only feasible with adequate habitat, good dispersal and colonization ability, availability of food resources, and absence of physical barriers that might preclude movements. Many species are confined to landscapes with no suitable transitory habitats that would allow them to respond quickly to the changing physical environment (Seavey, Gilmer, and McGarigal, 2011).

At the community level, alteration of food web structures, competition structure (including invasive species), and introduction of disease may result in changes to community assemblages and demographics, habitat structure, and dietary resources (Altizer, Bartel, and Han, 2011; Laaksonen *et al.,* 2010; Lafferty, 2009; Parmesan, 2006; Stralberg *et al.,* 2009). This reorganization of biological communities may create a mismatch in food resources, enable invasive species to expand, initiate trophic cascades, or cause local extirpations (Gilg, Sittler, and Hanski, 2009; Poloczanska *et al.,* 2008; Pounds *et al.,* 2006). Ultimately, adaptation may result in new biotic communities with no modern analog (Schneider and Root, 2002; Winder and Schindler, 2004). Wildlife demographic responses may include alterations in social groups, reproductive success, and age or sex ratios. For example, while reproduction rates for the tufted puffin (*Fratercula cirrhata*) were positively correlated with increasing sea-surface temperatures, growth rates and juvenile survival were reduced (Gjerdrum *et al.,* 2003). Fuentes *et al.* (2010) found that sea-level rise and storm events increased vulnerability of sea turtle (*Chelonia mydas*) rookeries to inundation and increased egg mortality, resulting in decreased reproductive success.

## EFFECTS ON SALT MARSH WILDLIFE

In salt marshes, wildlife habitat diversity is low because of the physiological conditions created by high salinity levels, tidal flooding, and low plant diversity. Fish, birds, marine mammals, several terrestrial mammals, and even a few reptiles such as the diamondback terrapin (*Malaclemys terrapin*) are found living in or near salt marshes, either as full-time residents or seasonal inhabitants. These species are adapted to survive in this dynamic tidal environment (Bias and Morrison 2006; Rush *et al.,* 2009; Spautz *et al.,* 2006; Tsao *et al.,* 2009). For example, the salt marsh sharp-tailed sparrow (*Ammodramus caudacutus*) selects its nesting period to avoid high spring tides, and the seaside sparrow (*Ammodramus maritimus*) chooses higher nesting locations (Gjerdrum, Elphick, and Rubega, 2005), both presumably to reduce impacts from tidal inundation on reproductive success.

However, many salt marsh vertebrates are listed as species of special concern or endangered species (USFWS, 1991, 2009;

IUCN, 2011) for which ecology is often little-studied and poorly understood. Mammals such as the Florida salt marsh vole (*Microtus pennsylvanicus dukecampbelli*) and the salt marsh harvest mouse (*Reithrodontomys raviventris*), which are both federally endangered species in the United States, have been impacted by habitat modification and loss of salt marshes (Shellhammer, 1982; Woods, Post, and Kirkpatrick, 1992). Salt marsh wildlife species are negatively affected by human actions that include modification and degradation of coastal processes through construction of barriers and alteration of natural hydrological flows (Brusati and Grosholz, 2009; Greenberg *et al.,* 2006; Nichols *et al.,* 1986; Schwarzbach, Albertson, and Thomas, 2006; Takekawa *et al.,* 2006).

Upslope movement of salt marsh is often restricted by barriers, or sediment is limited by hydrologic modifications (Rybczyk and Cahoon, 2002). Artificial levees, dikes, and sea walls may be obstacles to dispersal for less mobile birds such as the California clapper rail (*Rallus longirostris obsoletus*), although not for more mobile birds such as Nelson's sharp-tailed sparrows (*Ammodramus nelson*) or short-eared owls (*Asio Flammeus*) (Eddleman and Conway, 1998). Along the North American Atlantic and Gulf Coasts, extensive alterations have occurred from ditching for mosquito control and overgrazing by invasive nutria (*Myocastor coypus*), muskrats (*Ondatra zibethicus*), and waterfowl (Evers *et al.,* 1998). Tidal salt marsh bird diversity decreases with increasing fragmentation of salt marshes (Benoit and Askins, 2002); for example, the size and proximity of salt marshes influence the number of bird species along the New England coast of eastern North America (Shriver *et al.,* 2010). Displaced species may suffer from increased competition and predation rates or be maladapted to their new environment. The synergistic effects of urbanization, degradation, disturbance, pollution, and invasive species introductions will exacerbate impacts from sea-level rise and increasing severe storms (Atwater *et al.,* 1979; Kennish, 2001; Nichols *et al.,* 1986) on wildlife (Daniels, White, and Chapman, 1993; Hughes, 2004).

The added impacts of climate change on salt marsh ecosystems may greatly increase threats to already vulnerable populations and species in urbanized estuaries (Ohlemuller *et al.,* 2008). Highly urbanized estuaries such as those on the Chesapeake Bay (Jantz, Goetz, and Jantz, 2005), Saint Lawrence River (Jean and Bouchard, 1991), San Francisco Bay (Nichols *et al.,* 1986), and Yangtze River (He *et al.,* 2007) will be highly susceptible. In general, we suggest that the most vulnerable salt marsh habitats are: (1) located where wildlife habitats are already heavily fragmented and altered, often in close proximity to urbanization; (2) bordered by sea walls, levees, or roads that limit transgression or animal movements into upland transitions; (3) supporting biotic communities that are endemic, rare, or threatened by other limiting factors; or (4) affected by combinations of these factors, where stressors are amplified by increased sea-level rise or severe weather events. Several summaries have reported on the effects of sea-level rise on salt marshes of the Atlantic and Gulf Coasts of North America, but few studies have summarized effects on areas with mixed semidiurnal tidal ranges, such as on the Pacific Coast, or have focused specifically on the endemic wildlife. Thus, we use a case study from San Francisco Bay in California, U.S.A., to illustrate some of the primary conservation concerns and challenges facing salt marsh wildlife with changing climate.

## SAN FRANCISCO BAY SALT MARSHES: A CASE STUDY

### San Francisco Bay Estuary

The San Francisco Bay Estuary supports one of the largest extents of salt marsh in western North America. In 1850, it covered an estimated 2200 km$^2$ (Atwater *et al.,* 1979), but fragmentation and modification through local- and watershed-scale land-use changes resulted in loss of >80% of historic marshes (Goals Project, 1999). This created a mosaic of highly fragmented salt marshes with surrounding areas composed of urban and agriculture lands. A large proportion of the remaining salt marshes is located in the northern reach of the estuary on San Pablo Bay, and nearly 85% of San Pablo Bay salt marsh habitats has been altered by human activities such as diking, mining, salt pond development, and farming (USFWS, 2007). In recent years, efforts have been made to reverse the habitat losses, and restoration projects in this subregion have encompassed over 6.0 km$^2$ (Kershner, 2010).

San Pablo Bay is strongly influenced by ocean tides, freshwater runoff from local creeks and rivers, and freshwater inflow from the Sierra Nevada Mountains through the Sacramento–San Joaquin Delta via Suisun Bay and the Carquinez Strait (Cloern *et al.,* 2010, 2011). It has a mixed semidiurnal tidal regime and a Mediterranean climate with cool, wet winters where most precipitation comes as rain or fog followed by warm, dry summers. At the watershed scale, freshwater input and suspended sediment supply are dominated by discharge from the Sierra Nevada snowpack augmented by river discharge from low-elevation watersheds (Cloern *et al.,* 2011).

Projected climate change alterations to this system include coastal flooding from sea-level rise, higher snow lines, and a decreased winter snowpack, resulting in decreased freshwater input in the summer and declining late season runoff (Cayan *et al.,* 2001, 2006; Dettinger *et al.,* 2004; Knowles and Cayan, 2002, 2004; Knowles, Dettinger, and Cayan, 2006). With seasonal changes in precipitation and snow cover, including more rain and earlier snowmelt in the winter, alterations in freshwater input and estuarine salinity levels will change both temporally and spatially (Cayan *et al.,* 2008b; Dettinger and Cayan, 1995) and could alter salt marsh habitats and vegetation composition. In addition, current hydrogeomorphic processes within the estuary will be altered through changing suspended sediment loads and accretional processes, which maintain marsh elevations (Cahoon, Lynch, and Powell, 1996; Culberson, Foin, and Collins, 2004; Patrick and DeLaune, 1990; Temmerman *et al.,* 2004; Watson, 2008). Projected increase in extreme storm events may cause coastal edge erosion and plant drowning, which will alter salt marsh stability (*e.g.,* resistance to shear stress) and increase susceptibility to scour (Zedler, 2010).

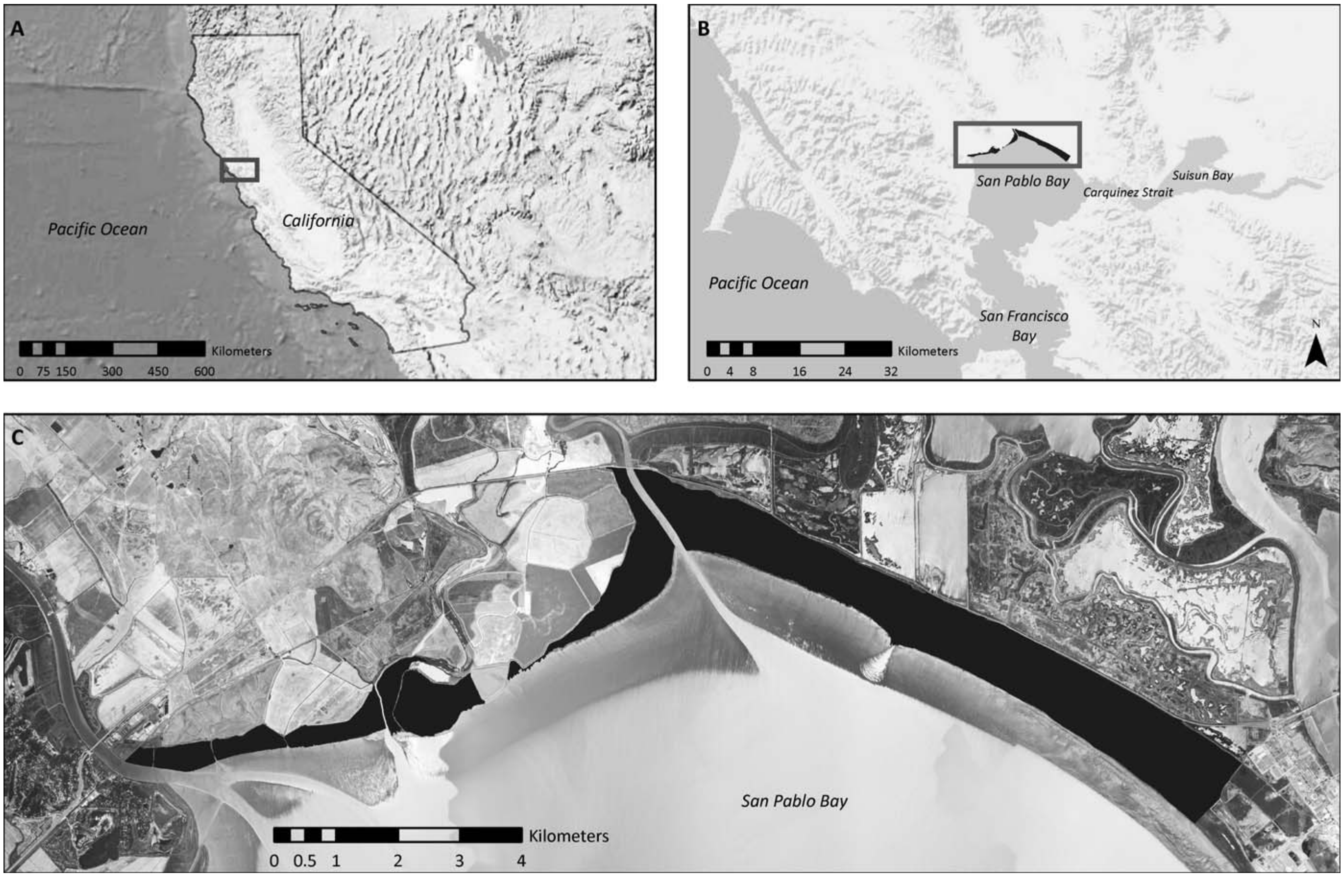


Figure 1. (A) San Pablo Bay National Wildlife Refuge in northern California, U.S.A. (B) San Pablo Bay National Wildlife Refuge is located in the upper reaches of the San Francisco Bay Estuary and receives water from the Sierra Nevada snowpack. (C) It spans most of the northern edge of the San Pablo Bay and is composed of open water, mud flats, and a large expanse of salt marsh.

## Refuge Salt Marshes

San Pablo Bay National Wildlife Refuge (hereafter SPBNWR; 38°3′9″ N, 122°24′22″ W) supports one of the largest, contiguous areas of remnant salt marsh in the estuary and hosts many endangered or threatened wildlife species (Figure 1). SPBNWR consists of a low-gradient salt marsh that rings San Pablo Bay with a mean elevation of 1.76 m NAVD88 and an elevation range of 1.49 m, where >88% of 950 elevation survey points fell within 1.5–2.0 m (Thorne *et al.,* 2010). Common pickleweed (*Sarcocornia pacifica*) dominates the large salt marsh platform (91% cover, 45 cm height) fringed by a narrow outer band (3.2% cover, 77 cm height) of Pacific cordgrass (*Spartina foliosa*) (Thorne *et al.,* 2010). The slight elevation gradient found at SPBNWR is characteristic of many salt marshes (Archibold, 1995), and is a critical geomorphic feature when considering salt marsh "drowning." The local current rate of sea-level rise in the estuary is 2.2 mm/y (Cayan *et al.,* 2008a), which will primarily affect the salt marsh over the long-term (>25 y), but may change vegetation inundation frequency over the short-term for most of the area. Over 30 y, there has been an observed change of vegetation composition in the San Francisco Bay estuary to more salt-tolerant species (*e.g., Sarcocornia pacifica*) (Watson and Byrne, 2012).

Mean high water (MHW) inundates the salt marsh platform at >1.68 m NAVD88, while most plants are inundated at mean higher high water (MHHW) at 1.85 m (Thorne *et al.,* written communication). During two storms in 2010 and 2011, the duration of salt marsh platform flooding increased significantly from nonstorm periods (2.7–3.8 h/d) to storm periods (6.7–8.3 h/d) (Figure 2). These storm events produced salt marsh platform inundation 1.8–3.1 times longer than during nonstorm periods within the same month. We found that during El Niño events, sea level increases with thermal expansion of seawater and prevailing winds, thus creating analogs for future sea-level rise conditions (Thorne *et al.,* written communication). In addition, increased frequency of these types of storms expected with climate change (Cayan *et al.,* 2008a) could increase salt marsh edge erosion and drowning of vegetation in the short-term. Salt marsh edge erosion is not currently occurring at SPBNWR because of the large amount of available sediment deposited during the Gold Rush era of the mid-1800s (Jaffe, Smith, and Foxgrover, 2007), which is sustaining the salt marsh edge. At the upland margin, salt marsh transgression upslope is limited by

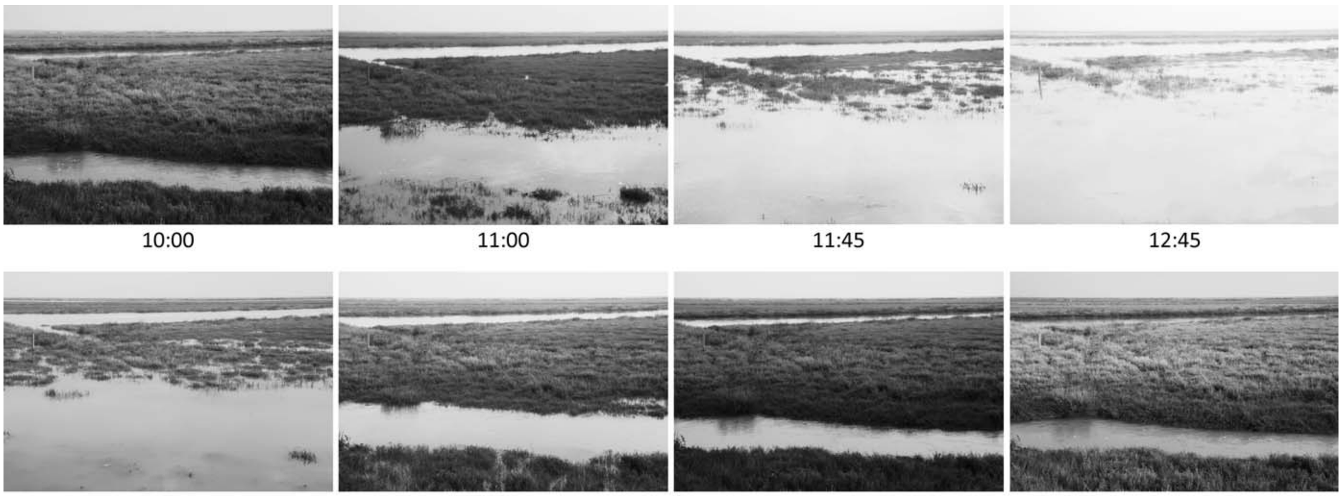


Figure 2. The San Pablo Bay National Wildlife Refuge contains large expanses of common pickleweed (*Sarcocornia pacifica*), which provides habitat and cover for endangered salt marsh wildlife species. These photos were taken on 29 January 2010 during a high tide concurrent with a low-pressure storm event. Small incremental increases in sea level or changes in the frequency of storms or depth of water result in elevated predation and drowning risk for many populations.

levees, a state highway, and private property. Therefore, sustaining the salt marsh through upslope migration over the long-term is currently not possible and will not likely occur without significant management actions, including the removal of highway infrastructure and work with local landowners.

## Effects on Wildlife

Eleven wildlife species of concern are named in the multispecies tidal marsh recovery plan for northern and central California (USFWS, 2009), and San Francisco Bay constitutes the single largest area of remnant salt marshes supporting them (Goals Project, 1999; Takekawa *et al.,* 2006). Many wildlife species, such as the San Pablo song sparrow (*Melodia melodia samuelis*), salt marsh common yellowthroat (*Geothlypis trichas sinuosa*), state-threatened California black rail (*Laterallus jamaicensis coturniulus*), and the federally endangered salt marsh harvest mouse, are local endemic species (Takekawa *et al.,* 2011) that reside within SPBNWR. The salt marsh harvest mouse and California black rail are closely associated with common pickleweed plants, which they use for feeding, nesting, and protection from predators (Bias and Morrison, 2006; Fisler, 1963; Tsao *et al.,* 2009). Takekawa *et al*. (2002) found a high correlation between salt marsh harvest mouse captures and height and percent cover of pickleweed plants.

Vegetation on the salt marsh platform at SPBNWR rarely becomes fully submerged except during tidal events that exceed mean higher high water (MHHW) or when they occur concurrently with storm-driven elevated sea levels (Figure 2). Over the short-term, greater inundation frequency and water depth may increase predation, decrease survival, and reduce reproduction. An 84% reduction in the amount of trapped salt marsh harvest mouse was observed in 2005–2006 after a large storm that submerged most of the salt marsh vegetation (Woo, Block, and Takekawa, 2008). These effects can be exacerbated when species are forced out of their normal salt marsh habitats to upland areas, which are typically marginal habitats. For example, the salt marsh harvest mouse has been shown to climb up common pickleweed plants or move to upland vegetation for refugia, but they are affected by predation by great egrets (*Ardea alba*) among others (Bias and Morrison, 1999; Hulst *et al.,* 2001; Johnston, 1956; Shellhammer, 1982), and competition from other small mammals such as house mice (*Mus musculus*). Non-native predators such as feral cats (*Felis catus*) and red foxes (*Vulpes vulpes regalis*) have been shown to prey on California clapper rails and light-footed clapper rails (*Rallus longirostris levipes*) and California black rails during high-water events (Evens and Page, 1986; Foin *et al.,* 1997; Harding, Doak, and Albertson, 2001).

Vegetation structure and cover provide safe places for salt marsh wildlife to nest and reproduce (Greenberg *et al.,* 2006). The California clapper rail nest is approximately located in vegetation 57 cm tall, often in cordgrass stems, presumably to reduce flooding risk while avoiding predation (Takekawa *et al.,* 2011). In addition, the California black rail has been shown to time breeding when inundation of the salt marsh is minimal (Takekawa *et al.,* unpublished data). Increased inundation frequency and depth also fill channel and tidal sloughs for longer periods of time, making them unavailable for terrestrial wildlife. The California clapper rail uses tidal sloughs and channels for movement, feeding, and escape from predators (Takekawa *et al.,* 2011).

Long-term impacts and changes to wildlife demographic and community structures from sea-level rise and storms are less clear than short-term impacts. However, long-term negative impacts to the overall amount and quality of salt marsh habitat will impact all salt marsh wildlife. For example, long-term salt marsh drowning and habitat loss will force wildlife to disperse to new habitats, exposing them to predation threats and new competitive interactions with other species. Species will not always be able to find suitable habitat within the range of their dispersal capability—for example, the California black rail was shown to only move 38 m during extreme high tides and 28 m on average (Tsao *et al.,* 2009). In addition, the California clapper rail, which prefers the low intertidal zone dominated by cordgrass for feeding and nesting (Foin *et al.,* 1997), will experience erosion of these edge habitats over the long-term (Cayan *et al.,* 2005). Long-term effects on local food availability, new competition structures, and predation rates could have coupled effects with "coastal squeeze" to result in extirpation or extinction (Geissel, Shellhammer, and Harvey, 1988).

SPBNWR will be affected by sea-level rise and storm events both over the short- and long-term (Table 1). It is a fragmented salt marsh in a highly urbanized estuary with barriers, including private property levees (*e.g.,* dikes protecting adjacent private agriculture land and highways), that inhibit marsh transgression upslope. The most likely future scenario will be narrowing and reduction of total geographical salt marsh area for many endemic and threatened species. Population viability modeling has indicated that the extent of salt marsh is the most important variable for predicting long-term viability of salt marsh species such as the San Pablo song sparrow (Takekawa *et al.,* 2006). Thus, we expect a decline in wildlife populations of SPBNWR, especially in regions with salt marshes that are already affected by stressors, unless proactive efforts are made to undertake adaptation actions. The loss of wildlife populations will occur in the long-term from sea-level rise effects but also in the short-term when more frequent extreme events will impact the salt marsh communities.

## CONSERVATION AND PLANNING CHALLENGES

Coastal land managers seeking to conserve salt marsh wildlife in light of climate change are faced with many challenges. First, there are many uncertainties about the magnitude and interactions of sea-level rise and storm frequency and intensity (Kettle, 2012). Second, most climate change models are developed at global or continental spatial scales, whereas most biological or ecological responses will be at local scales (Parmesan, Root, and Willig, 2000). This disconnect creates many challenges for local-level planning and adaptation strategies. In addition, life-history requirements, food web characteristics, and biotic interactions are not well understood for many tidal salt marsh species (Greenberg *et al.,* 2006). Reorganization of biotic communities will create uncertainty about future relationships among species in salt marshes, making predictions with climate envelope models of limited value. Lastly, the salt marsh resiliency to keep up with sea-level rise through accretion processes or transgression is not well known for many areas. To better understand ecomorphodynamic feedbacks in salt marshes, availability of suspended sediment, primary productivity, and organic inputs need to be studied and monitored.

Coastal planning that implements adaptive management including short- and long-term effects should be encouraged (*e.g.,* USFWS, 2009). In light of stochastic storm events and increased inundation frequency from sea-level rise, key short-term planning needs for salt marsh wildlife include increasing connectivity of habitats, monitoring wildlife population abundance and distribution, and ensuring the presence of upland refugia (either natural or constructed) for survival during flood events (Hannah and Hansen, 2005). We recommend that long-term planning needs address redesigning protected areas to plan for vegetation and habitat shifts, transitional and corridor habitats, marsh transgression, and restoring more areas to tidal flows to support wildlife in the short-term. For many species, detailed landscape-level population viability modeling to predict the probability of extinction or estimate minimum viable population numbers will be critical to make informed decisions.

Coordination of conservation planning strategies is beginning at multiple geographical scales; however, it is important to incorporate wildlife needs into these strategies (*e.g.,* U.S. Fish and Wildlife Service, Landscape Conservation Cooperative). Current protected areas would benefit from individual assessment of their resiliency, including areas like SPBNWR. Coastal plans that encourage vulnerability and connectivity assessments, as well as including future acquisitions and management actions that incorporate transition and future transgression zones, are important to avoid population extirpation or extinction. Conservation plans would benefit from incorporation of multiple stakeholders (including private property owners) and scientists to assess availability of salt marshes as future wildlife habitats. Innovative and novel methods to mitigate sea-level rise and storm impacts should be encouraged, such as sediment watershed management and use of dredge material to assist in salt marsh accretion (Ford, Cahoon, and Lynch, 1999; Yozzo, Wilber, and Will, 2004).

## CONCLUSIONS

In light of global change, it has become more apparent that salt marsh species reside in a dynamic ecosystem, the future of which is threatened. It will be the task of coastal scientists and managers to determine which climatic biophysical variables will limit their populations over the short- and long-term and which salt marsh areas are most at risk. Species most vulnerable to climate change are located in areas where wildlife habitats are already heavily fragmented and altered by human actions, lacking upland areas for salt marsh transgression or animal dispersal. Planning processes that identify and incorporate local climate change risks may improve the resiliency of salt marsh wildlife and reduce the risk of irreversible impacts. Regional conservation planning should be coupled with targeted research to conserve wildlife biodiversity as well as ecosystem functions and services of these salt marsh ecosystems.

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